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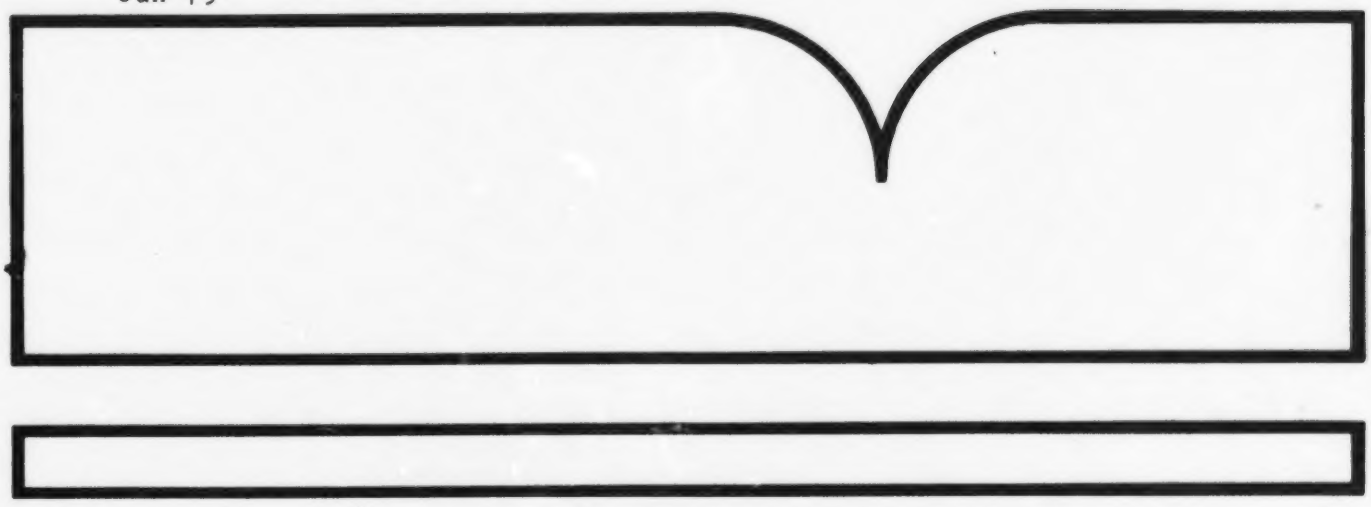

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16. ABSTRACT

This literature review summarizes current data on the effects of pesticides on marine organisms, aquatic environmental research methods, bioaccumulation of pesticides by estuarine and marine organisms, and biota residue. Results of studies of the environmental effects of dredging and the occurrence of diseases and tissue abnormalities in fish, shellfish, and marine plants also are reported.

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JOURNAL ARTICLE
Literature Review

Marine and estuarine pollution

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The state of pollution of the Mediterranean Sea was presented by Osterberg and Keckes.[1] Summary statements on pollution from metals, pesticides and PCBs, wastewater, oil, thermal discharge, and radioactivity were given; the authors concluded that wastewater and oil pollution can cause the greatest effect, especially near populated land masses. The estimated annual pollution loads of the Mediterranean Sea were presented; the majority of the wastes enter the sea via the rivers.[2]

In attempts to evaluate the economic impact of pollution on Atlantic Ocean and Gulf of Mexico fisheries, Kumpf [3] considered oil pollution, fish mortalities from effluents, habitat deterioration, and closure of pelecypod beds; the author concluded that habitat destruction greatly affected those species that used estuaries as nurseries.

Symposia proceedings and treatises on marine biology continue to be a source of new information and a synthesis of existing data on marine pollution.[4-13]

PESTICIDES

Livingston [14] reviewed the literature on the impact of pesticides on aquatic environments; estuaries and oceans were included in a discussion of the sources, routes of contamination, and fate of pesticides in aquatic systems.[15]

The responses of penaeid shrimp of the Gulf and South Atlantic region of the U. S. to toxic substances, including organochlorine, organophosphorous, and carbamate pesticides,[16] and toxicity data (test results, test organisms, test types, and experimental conditions) for the herbicides acrolein, dalapon, dichlobenil, diquat, and endothal were provided.[17]

Specific laboratory microcosms developed with pesticides to determine the impact of pollutants on several estuarine components were discussed by Wilkes.[18, 19] Hansen [20] reviewed the impact of pesticides on the marine environment as evidenced by results from laboratory bioassays. The use of biological indicator organisms to monitor the comparative pollution of aquatic areas (including estuarine ecosystems) by organochlorine pesticides was reviewed.[21]

Recent material on the effects, transformation, and degradation of commonly used pesticides in the environment was reviewed by Mel'nikov *et al.*[22] The effects of organochlorine compounds, principally pesticides, on marine birds were discussed.[23]

The Environmental Protection Agency (EPA), Environmental Research Laboratory, Gulf Breeze, Fla., described research conducted during 1977 on pesticide fate and effects in the estuarine environment.[24]

Methods. Hodgson *et al.*[25] described some

of the recently developed methodology for analyzing Kepone in finfish, shellfish, and oyster tissue. The Environmental Protection Agency's Ocean Disposal Bioassay Working Group [26] published a manual of bioassay procedures suitable for toxicity evaluations of waste materials (including pesticides) considered for ocean disposal under EPA's Ocean Disposal Permit Program. Procedures of various complexity utilizing several organism types were described.

Acute effects. The acute effects of pesticides on estuarine and marine organisms are presented in Table I.[27-32]

Sublethal effects. Rawls [33] investigated the shell regrowth of the eastern oyster (*Crassostrea virginica*) exposed to 2,4-D BEE in Maryland tidewaters and found that application rates of 20, 30, or 22.5, 33.75, or 67.5 kg/ha produced no effect. The author concluded that 2,4-D BEE could be used safely to clear oyster beds of watermill foil prior to fall oyster harvest.

Sheepshead minnows (*Cyprinodon variegatus*) exposed to chlordane concentrations ≥ 1.7 µg/l caused significant mortality of second-generation fish.[29] Exposure of sheepshead minnows to 4.8 µg/l or less of trifluralin significantly reduced fecundity of parental fish. Pentachlorophenol concentrations ≥ 195 µg/l significantly reduced both the hatch of sheepshead minnow embryos spawned by parental fish and the survival rate of second-generation fish.

Antipredator behavior of the grass shrimp *Palaemonetes pugio* was impaired by exposure to sublethal concentrations of methyl or ethyl parathion.[34] The shrimp became more susceptible to predation by gulf killifish, *Fundulus grandis*. The killifish were then provided a prey choice between the grass shrimp and the sheepshead minnow, both unexposed and exposed to methyl parathion. A greater proportion of grass shrimp were consumed when the prey were exposed.[35]

Densities of hard clams (*Mercenaria mercenaria*) in wild populations were seven to eight times as great in sections of shellfish beds where predators were reduced by Sevin and o-dichlorobenzene applications compared with untreated sections of the same beds.[36]

There was no significant change in the population density of the salt-marsh quail *Melampus bidentatus* in response to multiple treatments of granular formulations of either of the organophosphorous insecticides temephos or chlorpyritos, whereas temephos emulsions produced a significant population density depression.[37]

BHC, DDT and other pollutants were found in the tissue of *Nibea mitsukurii* exhibiting neoplastic disease occurring along the coastal area of southwest Japan.[38]

Daily additions of 10 µg chlordane to 1 l of dialysis culture, consisting of mixed species of estuarine phytoplankton, reduced algal growth rate and ^{14}C-uptake per unit of chlorophyll *a*.[39] Magnani *et al.*[40] found that chlordane and heptachlor at 50 µg/l reduced cell density, chlorophyll *a* per unit volume of culture, ^{14}C uptake per cell, and carbon fixation per unit of chlorophyll *a* in the marine dinoflagellate *Exuviella baltica*. The concentration of chlorophyll *a* per cell was not reduced, however, and chlordane was more toxic than heptachlor.

Chlordane inhibition of an extracellular endopeptidase was determined by Nakas,[41] who also found that protein and RNA synthesis were not inhibited.

Peddicord and McFarland [42] investigated the effects of suspended dredged material (containing chlorinated hydrocarbons among other pollutants) on various estuarine organisms.

Goodman *et al.*[43] conducted 28-day flow-through bioassays of heptachlor and toxaphene, using sheepshead minnow (*Cyprinodon variegatus*) embryo and fry. At heptachlor concentrations of 4.3, 3.5, 2.2, 2.0, and 1.2 µg/l, test animal survival was 1, 5, 61, 79, and 88%, respectively. Toxaphene concentrations of 2.5, 1.1, 0.6, 0.3, and 0.2 µg/l produced test animal survivals of 10, 85, 79, 88, and 80%, respectively.

Captan, at 30 µg/l, did not affect the survival of Dungeness crab (*Cancer magister*) zoeae in a 69-day exposure, although some molting delay did occur.[44] It was concluded that because of its low toxicity and high degradiation rate (half-life = 23 to 54 hours), captan applications near marine or estuarine waters were not likely to adversely affect crab populations. The toxicity of nine pesticides to various life-history stages of the Dungeness crab, *Cancer magister*, was examined by Caldwell,[45] who found that the zoeal stages were the most sensitive in long-term tests, approximately 5 to 10 and 10 to 100 times more sensitive than juvenile and adult crabs. The maximum acceptable toxicant concentrations (MATC) for *C. magister* zoeae were as follows: methoxychlor, 0.005 µg/l; chlordane, 0.015 µg/l; malathion, 0.02 µg/l; carbofuran, 0.05 µg/l; captan, 2 µg/l; DEF, 4 µg/l; trifluralin,

TABLE I. Acute toxicological effects of pesticides.

Compound	Group	Species	Type	Effective Concentration (µg/l)	Temp (°C)	Salinity (ppt)	Reference
Abate	Eel	*Anquilla japonica*	72-h LC_{50}-S	7 500	30 ± 2	33	27
	Fish	*Mugil cephalus*	72-h LC_{50}-S	600	30 ± 2	33	27
	Fish	*Mugil carinatus*	72-h LC_{50}-S	23	30 ± 2	33	27
	Decapod	*Metapenaeus monoceros*	72-h LC_{50}-S	45	30 ± 2	33	27
	Decapod	*Penaeus monodon*	72-h LC_{50}-S	45	30 ± 2	33	27
BHC	Decapod	*Penaeus duorarum*	96-h LC_{50}-FT	0.34	21–27	18–22	28
	Fish	*Lagodon rhomboides*	96-h LC_{50}-FT	86.4	22–26	20–23	28
Chlordane	Fish	*Cyprinodon variegatus*	96-h LC_{50}-FT	12.5	28	26	29
Dowicide G	Fish	*Cyprinodon variegatus* 2-wk fry	96-h LC_{50}-FT	516	30	24	30
	Fish	*Lagodon rhomboides* 48-h prolarvae	96-h LC_{50}-S	66	20	26	30
Halowax 1000	Decapod	*Palaemonetes pugio* postlarvae	96-h LC_{50}-S	440	19–24	20	31
Halowax 1013	Decapod	*Palaemonetes pugio* postlarvae	96-h LC_{50}-S	74	19–24	20	31
Halowax 1099	Decapod	*Palaemonetes pugio* postlarvae	96-h LC_{50}-S	69	19–24	20	31
Lindane	Mysidacea	*Mysidopsis bahia*	96-h LC_{50}-FT	6.28	23–25	15–22	28
	Decapod	*Penaeus duorarum*	96-h LC_{50}-FT	0.17	24–26	20–22	28
	Decapod	*Palaemonetes pugio*	96-h LC_{50}-FT	4.44	23–27	18–22	28
	Fish	*Cyprinodon variegatus*	96-h LC_{50}-FT	103.9	24–28	10–18	28
	Fish	*Lagodon rhomboides*	96-h LC_{50}-FT	30.6	22.5–25	21–23	28
Na-PCP	Decapod	*Palaemonetes pugio* 24-h larvae	96-h LC_{50}-S	649	25	24	31
	Pelecypod	*Crassostrea virginica* embryo	48-h EC_{50}-S	40	25	17	31
	Fish	*Lagodon rhomboides* 48-h prolarval	96-h LC_{50}-S	38	20	26	31
Na-PCP	Pelecypod	*Crassostrea virginica*	96-h EC_{50}-FT	76.5	8.4	20.3	32
	Decapod	*Penaeus aztecus*	96-h LC_{50}-FT	>195.0	25.0	26.5	32
	Decapod	*Palaemonetes pugio*	96-h LC_{50}-FT	>515.0	24.8	24.3	32
	Fish	*Fundulus similis*	96-h LC_{50}-FT	>306.0	24.4	22.9	32

TABLE I—(*Continued*)

Compound	Group	Species	Type	Effective Concentration (µg/l)	Temp (°C)	Salinity (ppt)	Reference
	Fish	*Lagodon rhomboides*	96-h LC_{50}-FT	53.2	25.0	20.8	32
	Fish	*Mugil cephalus*	96-h LC_{50}-FT	112.1	24.7	25.5	32
PCP	Fish	*Cyprinodon variegatus*					
		1-day fry	96-h LC_{50}-S	329	30	10	31
		6-wk fry	96-h LC_{50}-S	223	30	10	
	Fish	*Cyprinodon variegatus*	96-h LC_{50}-FT	442	29	24	29
Trifluralin	Fish	*Cyprinodon variegatus*	96-h LC_{50}-FT	190	29	21	29

15 µg/l; popanil, 80 µg/l; and 2,4-D, 1 000 µg/l.

An ultrastructural study of the mitochondria-rich "chloride" cells from the gills of seawater-adapted cichlid fish *Tilapia aurea* subjected to DDT showed no detectable significant ultrastructural effects.[46] Inhibition of intestinal water and amino acid transport and NA, K-APTase activity in killifish exposed to DDT *in vitro* were demonstrated.[47] Peters and Weber [48] determined the effect of acclimation history on the neural response of steelhead trout (*Salmo gairdneri*) to DDT and the effect of DDT on lateral line sensitivity.

The effects of DDT and Mirex, alone and in combination, on the salt-marsh fish *Adinia xenica* were determined by Koenig,[49] who found that at a high lethal level of DDT (54 µg/g·9 d) there was no increase in embryo or larval toxicity with increase in sublethal Mirex dose fed to parental females. The addition of the lowest dose of Mirex to the high lethal DDT dose, however, caused a significant increase in toxicity, as compared with the DDT dose alone.

Acetylcholinesterase inhibition in fishes exposed to five carbamate pesticides was similar (77 to 89%) regardless of species or exposure period (4 to 48 hours).[50] The author concluded that mechanisms related to injury of fishes by carbamates are quantifiable in the environment and should be measured in addition to chemical residues.

Bookhout and Monroe [51] found that the blue crab *Callinectes sapidus* could develop normally in malathion concentrations ranging from 0.02 to 0.11 mg/l, whereas the mud-crab *Rhithropanopeus harrisii* could develop only in concentrations ranging from 0.011 to 0.02 mg/l.

The mud-crab *Rhithropanopeus harrisii* exposed to the juvenile hormone mimic ZR-515 at levels of 0.01, 0.1, and 1.0 mg/l produced a significant reduction in survival of zoeal larvae with increasing concentrations in nearly all combinations of salinity and temperature.[52] The compound ZR-512 also affected mud-crab megalopa larvae survival and duration of larval development.[53]

Bioaccumulation. Results of studies on the bioaccumulation of pesticides by estuarine and marine organisms are presented in Table II.[54, 55]

Microbiology. Bourquin *et al.*[56] determined that Kepone concentrations as low as 0.02 mg/l inhibited growth and oxygen uptake of estuarine microorganisms. Malathion introduced to laboratory salt-marsh ecosystems at a field application rate of 420 g/ha was rapidly degraded by the indigenous bacterial community without apparent adverse effects on the microbial community.[57]

Metabolism. Kahn *et al.*[58] reviewed the present state of knowledge of the metabolism of pesticides by aquatic vertebrates and invertebrates, including *in vivo* and *in vitro* metabolic pathways of organophosphates, organochlorines, carbamates, and mercurials. James and his coworkers [59] studied xenobiotic metabolism in a number of marine fish and crustacea, including oxidative metabolism by the cytochrome P-450-dependent mixed-function oxidases and metabolism of several oxidation products. They concluded that if the same, or similar, pathways of biotransformation are utilized by marine and mammalian

TABLE II. Bioaccumulation of pesticides by estuarine and marine organisms.

Compound	Group	Species	Exposure	Bio-accumulation	Ref-erence
BHC	Decapod	*Penaeus duorarum*	0.19–0.68 μg/l: 96 h	51–142 X	28
	Fish	*Lagodon rhomboides*	32–91.3 μg/l: 96 h	308–554 X	28
DDD	Pelecypod	*Mytilus edulis*	0.05–2.18 μg/l: 50 h	9 120 X	54
Dieldrin	Pelecypod	*Crassostrea virginica*	1–100 μg/l: 43 d	5 100–5 500 X	55
	Pelecypod	*Mytilus edulis*	0.17–1.97 μg/l: 50 h	1 570 X	54
Endosulfan	Pelecypod	*Mytilus edulis*	0.14–2.05 μg/l: 50 h	600 X	54
Endrin	Pelecypod	*Mytilus edulis*	0.17–1.78 μg/l: 50 h	1 920 X	54
Halowax 1000	Decapod	*Palaemonetes pugio*	40 μg/l: 3 d	63 X	31
Halowax 1013	Decapod	*Palaemonetes pugio*	40 μg/l: 3 d	187 X	31
Halowax 1099	Decapod	*Palaemonetes pugio*	40 μg/l: 3 d	257 X	31
α-HCH	Pelecypod	*Mytilus edulis*	0.89–2.20 μg/l: 50 h	106 X	54
γ-HCH	Pelecypod	*Mytilus edulis*	1.24–2.29 μg/l: 50 h	100 X	54
Heptachlor	Fish	*Cyprinodon variegatus*	1.22–4.3 μg/l: 28 d	3 700–4 600 X	43
Heptachlor-epoxide	Pelecypod	*Mytilus edulis*	0.22–1.95 μg/l: 50 h	1 700 X	54
Lindane	Decapod	*Panaeus duorarum*	0.13–0.62 μg/l: 96 h	32–143 X	28
	Decapod	*Palaemonetes pugio*	1.0–5.5 μg/l: 96 h	25–80 X	28
	Fish	*Lagodon rhomboides*	18.4–31.3 μg/l: 96 h	167–287 X	28
	Fish	*Cyprinodon variegatus*	41.9–108.7 μg/l: 96 h	337–727 X	28
Na-PCP	Decapod	*Penaeus aztecus*	14–195 μg/l: 96 h	0.11–0.45 X	32
	Decapod	*Palaemonetes pugio*	32–515 μg/l: 96 h	0.5–3.0 X	32
	Fish	*Fundulus similis*	36–306 μg/l: 96 h	12–41 X	32
	Fish	*Mugil cephalus*	26–308 μg/l: 96 h	6.3–79 X	32
Toxaphene	Fish	*Cyprinodon variegatus*	0.20–2.5 μg/l	6 100–14 400 X	43

species, marine species metabolize and excrete xenobiotics much more slowly than the mammalian species.

The amphipod *Gammarus duebeni* exposed to sublethal concentrations of methylmercury of lindane in 2% sea water for 1 to 4 days showed significant reductions in hemolymph sodium concentration.[60] Prior treatment with sublethal concentrations of methyl mercury or lindane resulted in delays of 80 and 120 minutes, respectively, in the time required for *G. duebeni* to increase active sodium uptake when transferred from 100% to 2% sea water.

Pillai and his coworkers[61] exposed the marsh grass *Spartina alterniflora* root to Atrazine for 2 days and followed its uptake and metabolism for 28 days thereafter. After 2 days, 90% of the ^{14}C-Atrazine had been absorbed and translocated to the shoots. The Atrazine was readily metabolized to chloroform, to aqueous solutions, and subsequently to insoluble substances.

Fate. Bourquin and Gibson[62] discussed the general topic of microbial degradation of halogenated hydrocarbons, including pesticides, in the marine environment. Picer *et al.*[63] examined the equilibrium distribution of DDT between sea water and several particle types and found that DDT existed as a colloid dispersion in the systems studied.

Toxaphene was found to degrade in a few days in both sterile and unsterile anoxic salt marsh sediments and in sand containing the Fe(II)/Fe(III) redox couple.[64] The ultimate breakdown and toxicity of the degradation products were not determined. Atrazine and Ametryn dissipated rapidly in sugarcane and pineapple soils, and neither was found normally in adjacent estuarine sediments.[65] Diuron, however, was found in nearly all estuarine sediment samples obtained at 6-month intervals.

The effects of temperature, salinity, and sterility on the degradation of malathion, parathion, diazinon, and methoxychlor in estuarine water under controlled laboratory conditions were investigated.[66] No significant differences between sterile and nonsterile treatments were observed for any of the insecticides studied, while elevated temperature increased degradation of malathion, parathion, methyl parathion, and diazinon. Salinity effects varied among the four organophosphates.

The interaction of the marine deutermycete *Zalerion maritimum* grown in a simulated marine environment containing aldrin and dieldrin, with and without glucose, was studied.[67] Aldrin did not appear to degrade via dieldrin, and dieldrin, more than aldrin, cometabolized with glucose.

Pesticide residues. Priebe and Karow examined 30 fish in the North Atlantic and only once did they find that the total DDT in the musculature exceeded 2.0 ppm. Analysis of mirex residues in estuarine animals, sediments, and waters collected from the Mississippi Gulf in 1972-74 showed the following concentration ranges: Seston, 200 to 3 000 ppb; mollusks, 36 to 500 ppb; fish, 0 to 259 ppb; sediment, 3 to 5 ppb; and water, 0 to 0.01 ppb.[69] Pesticide residues in the range of 0 to 93 μg/kg dry matter were found in the marine algae *Sargassum thunbergii, Codium fragile,* and *Enteromorpha linza* sampled in southern Korean coastal waters during 1974 and 1975.[70]

Concentrations of organochlorine pesticide residues found in North Atlantic fish and shellfish were reported.[71, 72] A Butler and Schutzmann summary of 1 524 analyses of juvenile fish collected semiannually in 144 estuaries nationwide from 1972 to 1976 was presented.[73] Results for 20 common pesticides were presented. Additional information on biotic residues is provided in Table III.[74-84]

Pentachlorophenol (PCP's). In the proceedings of a symposium[85] on the chemistry, pharmacology, and environmental toxicology of the chemical pentachlorophenol, Brannon and Conklin[86] reported that exposure of the grass shrimp *Palaemonetes pugio* to sodium pentachlorophenate (Na-PCP) led to an apparent increase in the dry weight of exuvia as well as an increase in the total quantity of calcium.

The effects of Na-PCP on oxygen consumption of *P. pugio* varied depending on the stage of the molt cycle exposed.[87] Na-PCP was found to cause a complete inhibition of regeneration, a delay of initiation of limb bud development, or a reduction of limb bud growth, depending on the exposure concentration.[88] Ultrastructural changes induced by Na-PCP in the grass shrimp relation to the molt cycle were reported.[89]

Changes in the level of PCP in the hemolymph and hepatopancreatic tissues of the blue crab *Callinectes sapidus* were studied.[90] Fox and Rao investigated the effects of Na-PCP *in vivo* and *in vitro* on certain hepatopancreatic enzymes in *C. sapidus.*

The toxicity and bioaccumulation of Na-PCP by several estuarine organisms were determined (Table I).[30, 32] *P. pugio* in later stages of the proecdysial period exhibited a greater sensitivity to Na-PCP in 96-hour bioassays than shrimp in the intermolt and early proecdysial stages of the molt cycle.[92] Na-PCP significantly affected the feeding activity of the lugworm *Arenicola cristata* at concentrations of 80, 156, and 276 mg/l.[93]

Cantelmo and Rao[94] noted marked changes in nematode species composition as well as shifts in nematode feeding types in flow-through aquaria exposed to 161 and 622 mg/l PCP. Tagatz and his coworkers[95] studied the development of macrobenthic communities in flow-through experimental aquaria and determined that 15.8 and 161 mg/l PCP caused a marked reduction in the number of individuals and species found. At a concentration of 5×10^{-3} *M*, PCP and DNP caused inhibition of oxygen consumption in the tissues of the blue crab *Callinectes sapidus.*[96]

RESIDUES AND RELATED TOPICS

The results of the first year's mussel watch in the U. S. were summarized[97] and evaluated.[98] Mussels and clams were analyzed from 107 localities from the Pacific, Gulf, and Atlantic coasts for heavy metals (Ag, Cd, Cu, Ni, Pb, and Zn), radionuclides (Am-241, Cm-242, 244, Cs-137, Pb-210, and Pu-239, 240), halogenated hydrocarbons, and petroleum hydrocarbons. No seasonal differences were noted. No zones of generally high levels of all pollutants were found; however, local hot spots of a single pollutant were measured. Oysters were more effective as accumulators of Ag, Cu, Ni, and Zn than mussels, but the reverse was found for mussels and lead. The data suggested that mussels acquire radionuclides from suspended particles rather than directly from the water.

Field studies were undertaken as a means of learning the mechanisms of uptake of metals by organisms.[99-103] The oceanic residence time of several metals was calculated from data collected in the St. Lawrence Estuary.[99] The residence time in the water ranged from 30 years for Fe to 92 000 years for Cd with the residence time related to the length of time the metal can remain in solution. Concentrations of Cd, Cu, Hg, Ni, Pb, and Zn were determined in sea water near several outfalls in the Puget Sound.[100] Only Pb and

TABLE III. Pesticide residues: biota.

Compound	Species	Location	Residues	Reference
Chlordane	Lobster (*Homarus americanus*)	Canadian East Coast	lipid: 0.078–0.100 μg/g	74
	Herring (*Clupea hareugus*)	Canadian East Coast	lipid: 0.039–0.114 μg/g	74
DDE	Euphausid (*Meganyctiphanes norvegica*)	Med. Sea	26 μg/kg	75
Dieldrin	Periwinkle (*Littorina littorea*)	Medway Estuary, Kent, U.K.	0.1–1.6 μg/g	76
	Shore Crab (*Carcinus maenas*)	Medway Estuary, Kent, U.K.	0.8–10.7 μg/g	76
	Flounder (*Platichthys flesus*)	Medway Estuary, Kent, U.K.	7.4–210.0 μg/g	76
	Harbour Seal (*Phoca vitulina*)	German North Sea	0.15–0.54 mg/kg (blubber)	77
DDT	Sea-skater (*Halobates sobrinus*)	Baja California	180–400 ng/g	78
	Mussel (*Mytilus edulis*)	NW Atlantic	0.015 μg/g	79
	Red Crab (*Geryon quinquedues*)	NW Atlantic	0.061 μg/g	79
	Cod (*Gadus morhua*)	NW Atlantic	0.024 μg/g	79
	Striped Bass (*Roccus saxatilis*)	NW Atlantic	0.63 μg/g	79
	"Biota"	Gulf of Mexico	11.4 μg/g	80
	Periwinkle (*Littorina littorea*)	Medway Estuary, Kent, U.K.	1.0–6.3 μg/g	76
	Shore Crab (*Carcinus maenas*)	Medway Estuary, Kent, U.K.	4.8–25.7 μg/g	76
	Flounder (*Platichthys flesus*)	Medway Estuary, Kent, U.K.	10.1–650.3 μg/g	76
	Fish	Baltic Sea	0.06–4.00 mg/kg	81
	Dover Sole (*Microstomus pacificus*)	Southern California Bight	13–36 mg/kg (with eroded fins 1.8–25 mg/kg (without eroded fins)	82
	Harbour Seal (*Phoca vitulina*)	German North Sea	1.15–4.26 mg/kg (blubber)	77
	Grey Seal (*Halichoerus grypus*)	Nova Scotia	14.0 μg/g	83
DDT,DDD,DDE	Fish Cod liver	Baltic Sea	8.4–34.0 mg/kg	84
DDT,DDD,DDE	Fish Cod liver oil	Baltic Sea	6.0–20.0 mg/kg	84
Lindane	Harbour Seal (*Phoca vitulina*)	German North Sea	0.29–0.36 mg/kg (blubber)	77

Cd were found to be elevated; however, the concentrations of these metals were from 1 to 3.6 times greater in recently deposited sediments as compared with those deposited 50 years ago. The concentration of Pb was found to be inversely related to the particle size.[101] Concentrations of Cd, Cu, Pb, and Zn in an Australian estuary with a strong halocline were similar in both the fresh water and marine components.[102] Bryan and Hummerstone [103] transferred specimens of the pelecypod *Scrobicularia* from clean to polluted English estuary and vice versa. They found that the concentrations of Cd, Co, Pb, and Zn in the tissues gradually approached those specimens living within the area, but some differences were noted even after a year.

Fourie and Peisach [104] demonstrated serious losses of Cd, Pb, and Se from tissues during dehydration of marine zoological material, but

no losses of Co, Cr, Fe, Mn, or Zn were measured. Fish feed pellets containing 1.0 μg/g methyl mercury for 7 weeks contained 0.9 g/g in the kidney and 0.25 to 0.4 in muscles, spleen, and brain; however, when the pellets contained 1.0 μg/g $HgCl_2$, the burden levels only reached 0.2, suggesting that this fish is able to excrete mercury in this form.[105] Cd was added to sea water in concentrations from 0 to 50 μg/l to determine the uptake of this metal in the shell of *Mytelus edulis*. The periostracum contained up to five times that amount present in either the calcite or aragonite calcium carbonate layers of the shell. Since the periostracum contains the highest percentage of organic material, Sturgesson[106] concluded that the retention of Cd is related to protein metabolism in the mussel. Electron microprobe X-ray analysis of tissues under the electron microscope gave direct evidence for compartmentation of Cu and Zn into separate membrane-limited vesicles in amoebocytes of the oyster.[107] Accumulation of Cd in mussels was found to be inversely related to the salinity of the sea water.[108] The pathway of lead in the mussel was followed by Schulz-Baldes.[109] Lead was taken up by the gills and viscera from the water, distributed by the blood, and finally stored in membrane-bound vesicles within the excretory cells of the kidney.

The concentration of heavy metals in marine organisms is summarized in Table IV.[110-171] Data on the analysis of marine organisms for the presence of metals and other elements are presented according to the element, group of organism, and geographical region.

OIL POLLUTION

Mounting concern over possible effects of petroleum on high-latitude ecosystems was evident through work presented in two separate treatises. Thirty-four articles were presented at a symposium on the recovery potential of oiled marine northern environments and were equally divided among the following three sessions: long-term fate of petroleum hydrocarbons after spills—compositional changes and microbial degradation; physiological stresses and response in chronically oiled organisms; and long-term effects of oil spills on marine intertidal communities.[172] The effects of petroleum on Arctic and subarctic marine environments and organisms were treated in two volumes. The first dealt with properties and analyses of petroleum in biotic and abiotic systems; inputs, transport mechanisms and observed concentrations of petroleum in the marine environment; and alterations in petroleum resulting from physiochemical and microbiological factors.[173] A second volume[174] presents chapters on acute toxic effects of petroleum to marine organisms; diseases, host disease resistance, and pathological effects of petroleum; accumulation and biotransformation of petroleum in marine organisms; sublethal biological effects on bacteria, algae, invertebrates and fish; effects on ecosystems; and effects on marine birds and mammals. Proceedings from an international workshop included 43 papers on the methodology of analysis, biotic and abiotic occurrence, physical/chemical and biological fate, as well as effects of petroleum hydrocarbons in the marine environment.[175] The fate and effects of petroleum on nearshore environments, particularly those of the northeast U. S. Atlantic seaboard were summarized in 10 separate papers.[176] Effects of oil on marine ecology were addressed in a review that emphasized modifications on benthic systems of the British Isles.[177]

Developments in techniques for the analysis of petroleum in marine samples were presented. Interlaboratory precision was only ± 25% for analyses of replicate sediment samples, using similar techniques (Hilpert *et al.*).[178] The need for liquid chromatographic clean-up prior to gas chromatography for trace-level determinations[179] and a simple method for the detection of down to 2 μg oil in sediments were reported.[180] Procedures for the analyses of $C_{12} - C_{17}$ sulfur containing petroleum hydrocarbons were presented by Nakamura and Kashimoto.[181] No significant linear or logarithmic relationship was observed when simultaneous analyses of algal and invertebrate samples by ultraviolet spectrophotometry and gas chromatography were compared.[182] Niskin bottles were found to be a potential source of petroleum-like ($C_{20} - C_{35}$) hydrocarbons (Zsolnay[183]). Efficiencies for various modes of extraction of alkanes from marine sediments were studied by Motohiro and Iseya.[184]

Distributions and fates of petroleum within abiotic components of the marine ecosystem were studied. Input to the Beaufort Sea was estimated by Mackay[185] at about 20 mg/$m^2 \cdot y$ or about one-fifth that of the Mediterranean. The mean concentration of $C_{12} - C_{36}$ alkanes in oilfield brine was about 2 mg/l, resulting in an input of some 200 g of alkanes per day.[186] Sixty percent of oil released under an ice flow was incorporated into fracture

TABLE IV. Chemical residues reported from marine organisms.

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
Aluminum	Pelecypods	4–400; France; 110
	Pelecypods	6–460; U.S. Pacific; 97
	Pelecypods	20–540; U.S. Gulf; 97
	Pelecypods	24–160; U.S. Atlantic; 97
Arsenic	Fish	0.3–320; Canadian Arctic; 111
	Fish	0.1–4.4[a]; Australia; 112
	Fish	2.5–142; Greece; 113
Barium	Pelecypods	+[b]; France; 114
Cadmium	Blue-green algae	4.73; Malaysia; 115
	Green algae	8.29–16.32; Malaysia; 115
	Green algae	7–13.3; Norway; 116
	Green algae	0.1–3.8; Korea; 117
	Brown algae	0.3–3.7; Korea; 117
	Brown algae	3–29; Norway; 116
	Brown algae	0.65–6.4; England; 118
	Brown algae	5.63–13.8; Malaysia; 115
	Red algae	3.74–13.17; Malaysia; 115
	Red algae	0.3–2.8; Korea; 117
	Red algae	0.14–629[c]; Japan; 119
	Pelecypods	1.5–2.4; Hong Kong; 120
	Pelecypods	<0.07–3.3; Rhode Island; 121
	Pelecypods	0.4–13; Scandinavia; 122
	Pelecypods	1.7–12.4; U.S. Pacific; 97
	Pelecypods	1.3–10.7; U.S. Gulf; 97
	Pelecypods	0.6–21.4; U.S. Atlantic; 97
	Pelecypods	0.03–2.15[a]; Mississippi; 123
	Pelecypods	<0.15–3.2; U.S. Atlantic; 124
	Pelecypods	0–7.28; Canadian Atlantic; 125
	Pelecypods	0.7–6.8; Italy; 126
	Pelecypods	0.12–39.8; England; 103
	Pelecypods	0.48–31; England; 118
	Pelecypods	0.21–0.29[a]; Italy; 127
	Pelecypods	0.2–0.27[a]; Italy; 128
	Pelecypods	<0.4–4.1; Lebanon; 129
	Pelecypods	0.04–5.2[a,b]; U.S. Atlantic; 130
	Gastropods	<0.4–4.7; Lebanon; 129
	Cephalopods	0.12[a]; Italy; 128
	Stomatopods	0.46–0.54[a]; Italy; 128
	Decapods	0.03; South Carolina; 131
	Decapods	0–0.05; Canadian Atlantic; 125
	Holothurians	0.5–4.2; British Columbia; 132
	Fish	0.003–0.15[a]; U.S. Pacific; 133
	Fish	<0.3–0.9; Greece; 113
	Fish	0.05–0.18[a]; Washington; 134
	Fish	0–270; New Jersey; 135
	Fish	1.4–4.2; England; 136
	Fish	0.9–16; Canadian Arctic; 111
	Fish	<0.01; South Carolina; 131
	Fish	0.11–0.39[a]; Italy; 128
	Fish	0.01–0.1[a]; Australia; 112
	Fish	<0.005–0.23[a]; England; 137
Cesium	Green algae	0.037–0.048; Japan; 140
	Brown algae	0.037–0.07; Japan; 140
	Pelecypod	0.023; Japan; 140
	Gastropod	0.03; Japan; 140
	Cephalopod	0.016; Japan; 140
	Stomatopod	0.028; Japan; 140
	Decapod	0.05; Japan; 140
	Fish	0.06–0.16; Japan; 140

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
Chromium	Blue-green algae	17.03; Malaysia; 115
	Green algae	±588; Malaysia; 115
	Green algae	0.7–9.7; Korea; 117
	Brown algae	0.6–8.8; Korea; 117
	Brown algae	±25.6; Malaysia; 115
	Brown algae	1.1–10; England; 118
	Red algae	0.3–3.6; Korea; 117
	Red algae	±42.1; Malaysia; 115
	Pelecypod	0.46–12.8; England; 118
	Pelecypod	0.57–3.56; England; 103
	Pelecypod	0–31.6; Italy; 126
	Pelecypod	2.1–3.9; Italy; 141
	Pelecypod	5; France; 114
	Pelecypod	3.1–14.2; Rhode Island; 121
	Pelecypod	<0.3–3.6; U.S. Atlantic; 124
Cobalt	Green algae	4.4–15.2; India; 138
	Green algae	0.9–1.7; Korea; 117
	Green algae	0.17–0.36; Japan; 140
	Brown algae	1.3–9.2; India; 138
	Brown algae	1.0–11.9; Iceland; 139
	Brown algae	1.8–9; England; 118
	Brown algae	0.12–0.43; Japan; 140
	Brown algae	0.9–2.3; Korea; 117
	Red algae	0.3–2.6; Iceland; 139
	Red algae	1.3–14.5; India; 138
	Red algae	1.7; Korea; 117
	Pelecypod	3–121; England; 118
	Pelecypod	0–156; England; 103
	Pelecypod	0–6.3; Italy; 126
	Pelecypod	0.23–0.25[a]; Italy; 127
	Pelecypod	0.19–0.28[a]; Italy; 128
	Pelecypod	0.6–6.1; Rhode Island; 121
	Pelecypod	0.64; Japan; 140
	Gastropod	0.27; Japan; 140
	Cephalopod	0.06; Japan; 140
	Cephalopod	0.15–0.42[a]; Italy; 128
	Stomatopod	0.23–0.31[a]; Italy; 128
	Stomatopod	0.16; Japan; 140
	Decapod	0.05; Japan; 140
	Fish	0.24–1.01[a]; Italy; 128
	Fish	0.02–0.63; Greece; 113
	Fish	0.007–0.02; Japan; 140
Copper	Blue-green algae	3.78; Malaysia; 115
	Green algae	±26.1; Malaysia; 115
	Green algae	3.8–30.2; India; 138
	Green algae	5–27; Korea; 117
	Green algae	12–14; Japan; 140
	Green algae	10–44.7; Norway; 116
	Brown algae	4–38; Korea; 117
	Brown algae	1.9–10; Japan; 140
	Brown algae	11–150; Norway; 116
	Brown algae	20–107; England; 118
	Brown algae	1.8–8.8; Iceland; 139
	Brown algae	4.5–49.7; Malaysia; 115
	Brown algae	3.2–21.1; India; 138
	Red algae	3.5–80.4; India; 138
	Red algae	±13.4; Malaysia; 115
	Red algae	3–80; Iceland; 138
	Red algae	8.9–26; Japan; 142

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Red algae	12–43; Korea; 117
	Pelecypods	8.4–20.2; Hong Kong; 120
	Pelecypods	2.27–3.2[a,c]; India; 143
	Pelecypods	1.2–702[a,c]; U.S. Atlantic; 130
	Pelecypods	<0.3–11; U.S. Atlantic; 124
	Pelecypods	2.3–29.6; Rhode Island; 121
	Pelecypods	4.3–660; U.S. Atlantic; 97
	Pelecypods	21–410; U.S. Gulf; 97
	Pelecypods	3.5–11; U.S. Pacific; 97
	Pelecypods	2; France; 110
	Pelecypods	0.6–2,395[a]; Scotland; 107
	Pelecypods	13–102; England; 118
	Pelecypods	3.3–33.7; Italy; 126
	Pelecypods	1.26–1.48[a]; Italy; 127
	Pelecypods	1.23–1.47[a]; Italy; 128
	Pelecypods	26.3–185.6; Lebanon; 129
	Gastropods	11.3–135; Lebanon; 129
	Gastropods	17; Japan; 140
	Cephalopods	15.8–29.2[a]; Italy; 127
	Stomatopods	5.0–27.9[a]; Italy; 127
	Stomatopods	35; Japan; 140
	Decapods	14; Japan; 140
	Decapods	0.01; So. Carolina; 131
	Holothurians	0.4–42; British Columbia; 132
	Fish	0.1–2.8[a]; Australia; 112
	Fish	0.51–1.6; Japan; 140
	Fish	1.1–220; Canadian Arctic; 111
	Fish	0.07–1.55[a]; Italy; 127
	Fish	1–35; Greece; 113
	Fish	80–3,700; New Jersey; 135
	Fish	0.2–23.4; Newfoundland; 144
	Seals	1.5–5.6; Japan; 145
Iron	Blue-green algae	946; Malaysia; 115
	Green algae	4 735–13 816; Malaysia; 115
	Green algae	353–1 293; India; 138
	Green algae	81–590; Japan; 140
	Green algae	404–3 160; Korea; 117
	Brown algae	11–2 817; Korea; 117
	Brown algae	35–430; Japan; 140
	Brown algae	348–15 474; Malaysia; 115
	Brown algae	130–1 796; India; 138
	Brown algae	401–1 300; England; 118
	Red algae	486–2 302; Malaysia; 115
	Red algae	130–1 029; India; 138
	Red algae	28–1 051; Korea; 117
	Red algae	125–650; Japan; 142
	Red algae	148–1 320; Japan; 119
	Pelecypods	695–3 975; Hong Kong; 120
	Pelecypods	150; Japan; 140
	Pelecypods	59–710; U.S. Pacific; 97
	Pelecypods	140–690; U.S. Gulf; 97
	Pelecypods	81–370; U.S. Atlantic; 97
	Pelecypods	214–790; Rhode Island; 121
	Pelecypods	14–1 366; Scandinavia; 146
	Pelecypods	441–2 280; England; 118
	Pelecypods	177–3 480; England; 103
	Pelecypods	40–2 000; France; 114
	Pelecypods	22.6–48.3[a]; Italy; 127
	Pelecypods	25.3–34.3[a]; Italy; 128
	Pelecypods	197–1 171; Lebanon; 129

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Gastropods	216–5 916; Lebanon; 129
	Gastropods	1 101; Japan; 140
	Cephalopods	16; Japan; 140
	Cephalopods	1.3–22.4[a]; Italy; 128
	Stomatopods	11.3–16.6[a]; Italy; 128
	Stomatopods	29; Japan; 140
	Decapods	76; Japan; 140
	Fish	6.1–23; Japan; 140
	Fish	2 500–60 000; New Jersey; 135
	Fish	0.3–0.63; So. Carolina; 131
	Fish	0.5–49.4[a]; Italy; 128
	Fish	15–530; Greece; 113
Fluoride	Pelecypods	138–312; Italy; 141
Lead	Blue-green algae	17; Malaysia; 115
	Green algae	13–58.8; Malaysia; 115
	Green algae	9.2–19.5; India; 138
	Green algae	2.1–10.7; Korea; 117
	Green algae	0.7–2.0[a]; Washington; 100
	Green algae	15–301.5; Norway; 116
	Brown algae	1.0–1.4[a]; Washington; 100
	Brown algae	2.8–8.9; Korea; 117
	Brown algae	5.2–49.7; Malaysia; 115
	Brown algae	3–34.7; India; 138
	Brown algae	5.9–109; England; 118
	Brown algae	10.4–288; Norway; 116
	Red algae	0.36–8.66; Japan; 142
	Red algae	0.9–4.1; Korea; 117
	Red algae	1.7–44.8; Malaysia; 115
	Red algae	6.1–197.5; India; 138
	Pelecypods	0–50; Italy; 126
	Pelecypods	0.64–3.29[a]; Italy; 127
	Pelecypods	0.71–0.93[a]; Italy; 128
	Pelecypods	1.0–14[a]; Italy; 147
	Pelecypods	38.3–68.9; Lebanon; 129
	Pelecypods	5–6; France; 114
	Pelecypods	0–1 130; England; 103
	Pelecypods	26–983; England; 118
	Pelecypods	3–664; Scandinavia; 122
	Pelecypods	11.1–29.4; Rhode Island; 121
	Pelecypods	<0.4–1.4; U.S. Atlantic; 124
	Pelecypods	<0.1–9.5; U.S. Atlantic; 97
	Pelecypods	0–3.5; U.S Gulf; 97
	Pelecypods	0–2.35[a]; Mississippi; 123
	Pelecypods	0.2–12; U.S. Pacific; 97
	Pelecypods	0.5–1.5[a]; Washington; 100
	Pelecypods	30.4–52.5; Hong Kong; 120
	Gastropods	<6.8–95.6; Lebanon; 129
	Cephalopods	12.3–26.3[a]; Italy; 128
	Stomatopods	14.9–25.2[a]; Italy; 128
	Holothurian	0.4–3.7; British Columbia; 132
	Fish	0.015–1.3[a]; Washington; 134
	Fish	0.1–1.2; Canadian Arctic; 111
	Fish	0.1–4.1[a]; Australia; 112
	Fish	0.43–2.82[a]; Italy; 128
	Fish	8.1–24.5; England; 136
	Seals	0.03–0.04; Japan; 145
Magnesium	Blue-green algae	7 095; Malaysia; 115
	Green algae	4 455–16 324; Malaysia; 115
	Brown algae	7 343–13 937; Malaysia; 115

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Red algae	4 121–41 117; Malaysia; 115
	Pelecypods	5–40; France; 114
	Fish	100–240; Newfoundland; 144
Manganese	Blue-green algae	123; Malaysia; 115
	Green algae	92–221; Malaysia; 115
	Green algae	72.4–2 686; India; 138
	Green algae	24–202; Korea; 117
	Green algae	18–26; Japan; 140
	Brown algae	4.9–41; Japan; 140
	Brown algae	16–290; Korea; 117
	Brown algae	33–531; India; 138
	Brown algae	90.2–284; Malaysia; 115
	Brown algae	85–393; England; 118
	Brown algae	13–680; Iceland; 139
	Red algae	8.6–100; Iceland; 139
	Red algae	25.1–3 421; India; 138
	Red algae	23.7–316; Malaysia; 115
	Red algae	15–80; Korea; 117
	Red algae	27.5–78; Japan; 142
	Red algae	3.02–1 800[c]; Japan; 119
	Pelecypods	12.5–67; Hong Kong; 120
	Pelecypods	7.7; Japan; 140
	Pelecypods	2.5–25; U.S. Pacific; 97
	Pelecypods	8.9–18; U.S. Gulf; 97
	Pelecypods	5.9–41.9; U.S. Atlantic; 97
	Pelecypods	38–159; Rhode Island; 121
	Pelecypods	18–183; England; 118
	Pelecypods	4–360; England; 103
	Pelecypods	6.2–91; Scandinavia; 146
	Pelecypods	5–6; France; 114
	Pelecypods	0–37.8; Italy; 126
	Pelecypods	0.96–1.89[a]; Italy; 127
	Pelecypods	1.15–1.53[a]; Italy; 128
	Cephalopods	0.26–0.65[a]; Italy; 128
	Cephalopods	1.4; Japan; 140
	Stomatopods	1.9; Japan; 140
	Stomatopods	0.52–0.81[a]; Italy; 128
	Decapods	0.25; So. Carolina; 131
	Decapods	1; Japan; 140
	Fish	0.31–0.79; Japan; 140
	Fish	20–300; New Jersey; 135
	Fish	0.02–0.12; So. Carolina; 131
	Fish	0.23–2.3[a]; Italy; 128
	Seals	42.9–62.4; Japan; 145
Mercury	Green algae	0.68; France; 148
	Green algae	0.004–0.008[a]; Washington; 100
	Brown algae	0.002–0.006[a]; Washington; 100
	Brown algae	3.96; France; 148
	Brown algae	0.0002–0.14; France; 149
	Red algae	0.05–6.7; France; 148
	Polychaetes	0.5–1.5; Georgia; 150
	Polychaetes	0.01–0.08[a]; Germany; 151
	Pelecypods	0.03–0.07[a]; Germany; 151
	Pelecypods	4.57; France; 148
	Pelecypods	0.009–0.38; Italy; 126
	Pelecypods	0.02; Italy; 152
	Pelecypods	0.032–0.227[a]; Italy; 127
	Pelecypods	0.029–0.176[a]; Italy; 128
	Pelecypods	0.02–0.339[a]; Italy; 147
	Pelecypods	0.96–1.35; Italy; 141

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Pelecypods	<0.05–0.36; Monaco; 153
	Pelecypods	0–0.19[a]; Mississippi; 123
	Pelecypods	0.8–2.5; Georgia; 150
	Pelecypods	3.2–120.1[a,c]; California; 155
	Pelecypods	0.009–0.017[a]; Washington; 100
	Pelecypods	<0.18; U.S. Atlantic; 124
	Gastropods	0.002–0.03; California; 154
	Gastropods	0.6–9.4; Georgia; 150
	Gastropods	0.36; France; 148
	Gastropods	0.06–0.18[a]; Germany; 151
	Cephalopods	0.06–0.6; Italy; 152
	Cephalopods	0.123–0.425[a]; Italy; 128
	Barnacles	0.22; France; 148
	Amphipods	0.01–0.15[a]; Germany; 151
	Isopods	0.22–0.56[a]; Germany; 151
	Stomatopods	0.119–0.56[a]; Italy; 128
	Decapods	0.04–1.1; Italy; 152
	Decapods	0.07–2.96; Monaco; 153
	Decapods	0.03–0.12[a]; Germany; 151
	Decapods	0.3–0.9; Georgia; 150
	Decapods	0.23[a]; So. Carolina; 131
	Decapods	0.008–0.52[a]; California; 154
	Sea urchins	0.016–0.05; California; 154
	Echinoderms	0.1; Georgia; 150
	Fish	0.9–2.58[a]; Washington; 156
	Fish	0.085–0.383[a]; U.S. Pacific; 157
	Fish	0.005–3.17[a]; California; 154
	Fish	0.12–0.6[a]; So. Carolina; 131
	Fish	0.3–42; Georgia; 150
	Fish	0.18–0.32; Delaware; 158
	Fish	<0.01–2.4; British Isles; 159
	Fish	0–2.5; Italy; 152
	Fish	0.012–0.5[a]; Italy; 128
	Fish	0.01–2.7; Finland; 160
	Fish	0.24–1.9; Greece; 113
	Fish	<0.02–1.94[a]; Australia; 112
	Fish	0.05–0.59; Pacific-Indian Oceans; 161
	Fish	0.4–7 482; Pacific-Indian Oceans; 162
	Fish	0.04–0.42; Pacific-Indian Oceans; 163
	Fish	0.07–14.2; Indian Ocean; 164
	Fish	0–0.29; Pacific Ocean; 165
	Fish	0.5–27.6[c]; Japan; 166
	Fish	0.09–0.64; Japan; 167
	Seals	0.14–510[a]; Finland; 168
	Seals	0.08–79.2[a]; Canadian Arctic; 169
	Marine mammals	0.31–1 343; France; 114
Nickel	Green algae	2.3–6.9; Japan; 140
	Green algae	7–39.1; India; 138
	Brown algae	0.5–18.3; India; 138
	Brown algae	2.1–4.6; Japan; 140
	Brown algae	0.7–4.8; England; 118
	Red algae	6–22; India; 138
	Pelecypods	1.43–43.1; England; 103
	Pelecypods	1.6–25; England; 118
	Pelecypods	2; France; 114
	Pelecypods	0–11; Italy; 126
	Pelecypods	0.84–0.92[a]; Italy; 127
	Pelecypods	0.74–0.98[a]; Italy; 128
	Pelecypods	0–98.4; Lebanon; 129

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Pelecypods	<0.4–1.2; U.S. Atlantic; 124
	Pelecypods	7.4–37.5; Rhode Island; 121
	Pelecypods	<0.1–6.3; U.S. Atlantic; 97
	Pelecypods	0.9–7.3; U.S. Gulf; 97
	Pelecypods	0.7–5.3; U.S. Pacific; 97
	Gastropods	0–75.9; Lebanon; 129
	Gastropods	2.4; Japan; 140
	Cephalopods	1.1; Japan; 140
	Cephalopods	0.4–1.12[a]; Italy; 128
	Stomatopods	0.48–0.83[a]; Italy; 128
	Stomatopods	1.2; Japan; 140
	Fish	0.31–0.55; Japan; 140
	Fish	10–460; New Jersey; 135
	Fish	0.43–1.8[a]; Italy; 128
Rubidium	Green algae	11–19; Japan; 140
	Brown algae	20–52; Japan; 140
	Pelecypods	4.2; Japan; 140
	Gastropods	3.2; Japan; 140
	Cephalopods	3.4; Japan; 140
	Stomatopods	3.2; Japan; 140
	Decapods	4.1; Japan; 140
	Fish	2.6–4; Japan; 140
	Fish	0.52–2.9; Greece; 113
Selenium	Fish	0.24–44.3; Pacific-Indian Oceans; 161
	Fish	<0.04–0.8[a]; Australia; 112
	Fish	0.75–6.6; Greece; 113
	Seals	0.06–170[a]; Finland; 168
	Seals	4.13–30.4[a]; Canadian Arctic; 169
	Marine mammals	477; France; 110
Silver	Brown algae	<0.1–.036; England; 118
	Pelecypods	<0.1–1.12; England; 118
	Pelecypods	0–1.58; England; 103
	Pelecypods	<0.1–0.7[a]; U.S. Atlantic; 124
	Pelecypods	0.02–2.3[a,c]; U.S. Atlantic; 130
	Pelecypods	1.16–4.64; Rhode Island; 121
	Pelecypods	0.04–18.9; U.S. Atlantic; 97
	Pelecypods	0.1–6; U.S. Gulf; 97
	Pelecypods	0–2.3; U.S. Pacific; 97
Strontium	Pelecypods	16–64; U.S. Pacific; 97
	Pelecypods	12–28; U.S. Gulf; 97
	Pelecypods	19–56; U.S. Atlantic; 97
Tin	Pelecypods	4–600; France; 114
	Fish	<0.02–0.04; Greece; 113
Vanadium	Tunicates	<1–4 500; British Columbia; 170
Zinc	Blue-green algae	38; Malaysia; 115
	Green algae	38.9–104; Malaysia; 115
	Green algae	2.8–23.8; India; 138
	Green algae	2–95; Korea; 117
	Green algae	19–170; Japan; 140
	Green algae	950–6 233; Norway; 116
	Brown algae	1 590–4 700; Norway; 116
	Brown algae	20–52; Japan; 140
	Brown algae	8–84; Korea; 117
	Brown algae	15.5–210; Malaysia; 115
	Brown algae	4.5–57.4; India; 138
	Brown algae	138–1 330; England; 118
	Brown algae	2.5–73; Iceland; 139
	Red algae	24–75; Iceland; 139
	Red algae	14.2–69.2; Malaysia; 115

TABLE IV—(*Continued*)

Element	Type of Organism	Concentration (μg/g dry weight), Locality, and Reference
	Red algae	3.7–204; India; 138
	Red algae	101–300; Japan; 142
	Red algae	12–68; Korea; 117
	Pelecypods	64.4–97; Korea; 117
	Pelecypods	14.1–55[a,c]; India; 143
	Pelecypods	160; Japan; 140
	Pelecypods	60–260; U.S. Pacific; 97
	Pelecypods	310–7 080; U.S. Gulf; 97
	Pelecypods	68–4 060; U.S. Atlantic; 97
	Pelecypods	145–276; Rhode Island; 121
	Pelecypods	2–75; U.S. Atlantic; 124
	Pelecypods	22.5–1 980[a,c]; U.S. Atlantic; 130
	Pelecypods	87–9 630; England; 103
	Pelecypods	501–11 400; England; 118
	Pelecypods	14.6–27.4[a]; Italy; 127
	Pelecypods	15.5–17.1[a]; Italy; 128
	Pelecypods	0–379; Italy; 126
	Pelecypods	2–10; France; 114
	Pelecypods	15–14 746; Scotland; 107
	Pelecypods	14–460; Scandinavia; 146
	Gastropods	33; Japan; 140
	Cephalopods	12.3–26.3[a]; Italy; 128
	Stomatopods	14.9–24.2[a]; Italy; 128
	Stomatopods	110; Japan; 140
	Decapods	65; Japan; 140
	Decapods	5.2; So. Carolina; 131
	Holothurians	108–375; British Columbia; 132
	Fish	45–98.1; England; 136
	Fish	5.6–41.5[a]; Italy; 128
	Fish	7–60; Greece; 171
	Fish	15–111; Greece; 113
	Fish	700–7 100; New Jersey; 135
	Fish	6–62; Newfoundland; 144
	Fish	0.02–0.6; So. Carolina; 131
	Fish	0.003–0.016[a]; U.S. Pacific; 133
	Fish	14–180; Canadian Arctic; 111
	Fish	16–39; Japan; 140
	Fish	0.5–24[a]; Australia; 112
	Seals	7.8–32.2; Japan; 145

Note: ± means at or near the given value.
[a] Wet weight.
[b] Element present, but no amount given.
[c] Laboratory study.

zones and onto ice surfaces after 4 days as a results of currents, wind, and buoyancy spreading.[187] Shaw [188] found pelagic tar in Alaskan waters that was generally more weathered, but less abundant, than that encountered in the north Atlantic. Petroleum hydrocarbons were more prevalent during winter months in Nova Scotia waters because of the increased use of heating oils.[189] Fluorescence spectrometry of water samples revealed concentrations of aromatic hydrocarbons decreasing with depth, with surface values of 22 to 40 g/l declining to 0.6 to 1.3 g/l at 15 m.[190] Marine waters from the Antarctic can contain only biogenic hydrocarbons, even though sediments below were found by Mackie *et al.*[191] to reflect a past history of severe contamination. Alaskan gas seeps can be of a magnitude such that ethane and butane were detected by Cline and Holmes [192] 100 km downcurrent. Nearshore marine sediments can accurately chronicle an area's history of hydrocarbon input.[193, 194] Solubilities for 10 polynuclear aromatic hydrocarbons in sea water were estimated by Rossi and Neff.[195] The physical impact and interaction of whole oil with beach sediments was

monitored following a 1976 grounding in Spain.[196]

Concentrations of hydrocarbons in marine organisms were reported. Biogenic and petroleum hydrocarbons were characterized in a brown alga [197] and marine angiosperms.[198] Mollusks, particularly bivalves, continued to be studied as biological indicators of water quality,[179, 181, 199-201] although they may not necessarily reflect the hydrocarbon composition of their immediate environment.[193] Contamination by alkanes of petroleum origin was noted in several genera of crustaceans.[202-204] Sediments and animals from a Mediterranean island contained fossil fuel hydrocarbons.[205] Fish and benthic invertebrates from waters adjacent to an inoperative whaling station showed little evidence of contamination by earlier discharges.[191]

Most of the research with marine bacteria dealt with some aspect of petroleum biodegradation. Depending on the source of sea water, it was estimated that only one-half of most crude oil is degradable.[206] Soya-lecithin and ethyl allophosphate were recommended as fertilizers for enhancing degradation.[207] The numerical taxonomy and nutrient ecology of petroleum degrading bacteria were studied in nature.[208-210]

Most components of No. 2 fuel oil slicks were found by Horowitz and Atlas[211] to degrade at similar rates, under simulated Arctic conditions. Laboratory-weathered oil was 3 to 30 times more toxic to species of *Serratia* than fresh oil extracts, but effects on these bacteria were reversible as shown by back inoculation.[212] A diversity of microflora was associated with instances of crude-oil-induced fin erosion in mullet; however, *Vibrio* type colonies were always present.[213] Gutnick and Rosenberg [214] discussed the interactions between microbes and hydrocarbons.

Investigators [215-217] studied the effects of oil exposure under field conditions. Separate studies pointed to the importance of baseline data in assessing discrepancies between initial and final observations of gross ecological impact. The variations in size classes of burrowing clams following a spill of No. 2 fuel oil and JP5 jet fuel were documented over a 6-year period by Dow.[218] Egg oiling studies conducted at nest sites produced results similar to those from laboratory work with eiders.[219] Platt [220] found a rapid recovery of an Antarctic benthic system following cessation of whaling operations, suggesting that low temperature environments may not be less able to recover from damage inflicted by oil discharges. Low level oil input was deemed responsible for increased colony mortality rates, decreased numbers of breeding colonies, ovaria, planulae larvae, and settlement rates in Red Sea *Stylophora* coral.[221] Russell and Carlson [222] chronicled the recovery of a coral atoll from effects similar to those observed with previous large spills; however, results were produced from observations on flora and fauna affected by a 1975 release of vegetable, not petrol-type, oil.

Numerous papers chronicled initial investigations on the massive *Amoco Cadiz* spill off the Brittany coast of France. An overview of a proposed 3-year effects study was outlined by Laubier [223]; effects studies were reported by other authors as well.[224-226] Large quantities of oil were accommodated by affected sediments,[227, 228] while oxidized sulfur derivatives appear destined for persistent appearances because of seasonal reshuffling of beach deposits.[229] An extensive hydrocarbon monitoring program is under way; initial results for aromatics in water samples and zooplankton were given.[230] Preliminary results suggested that the benthic amphipod (*Ampelisca*) populations have been modified,[231] though intertidal surveys indicated localized, selective damage, with algal communities often recovering rapidly.[232] Early surveys revealed auks as the most abundant casualty among avian fauna.[233]

Laboratory studies generally reaffirmed previous trends regarding the toxicity of petroleum hydrocarbons to marine organisms; researchers tended to focus on specific hydrocarbons rather than oil mixtures. Ephemeral aromatic hydrocarbons such as naphthalene and trimethylbenzene were more acutely toxic to amphipods (*Elasmopus*) than the more persistent o-cresol or o-toluidine.[234] Triaromatic hydrocarbons were generally more toxic than diaromatics (naphthalenes) to a polychaete.[195] The toxicities of toluene and No. 2 fuel oil were found by Blundo [235] to be greater to barnacle larvae than to naphthalene, yet somewhat lower to benzene. For a series of naphthalenes, successive alkyl substitutions resulted in twofold increases in acute toxicity to calanoid copepods.[236] Data on the toxicity of naphthalenes for shrimp were presented.[237] No. 2 fuel oil was of low toxicity to the sea louse, *Lygia*[238]; Kuwait crude oil toxicity to estuarine phytoplankters was directly related to temperature and contact time.[239] Results of tests with four crude oils, five fuel oils, and

three strains of microalgae suggested that fewer water soluble components with boiling points higher than phenanthrene may account for much of the toxicity elicited.[240] Exposure to sublethal levels of No. 2 fuel oil for three generations produced successive increases in juvenile polychaete resistance to petroleum.[241] Variables affecting No. 2 fuel oil toxicity were discussed by Michael and Brown.[242] Diesel oil was more toxic to Arctic amphipods than Prudhoe crude; asphaltic residues of crude products were nontoxic.[243] Acrylic fibers decreased effluent hydrocarbon levels by one order of magnitude for bioassay waters flowing at rates of up to 20 ml/min.[244]

Dispersant toxicity to marine life continued to be examined. Standard toxicity tests were developed for open water and beach applications and the results discussed by Blackman *et al.*[245] relative to the licensing of dispersants in the United Kingdom. New generation plastic absorbents act as nontoxic inert substances that can ameliorate effects of oil alone.[246] In tests with an intertidal red algae and 10 emulsifiers and surfactants, the ester-type polyoxyethylene sorbitan trioleate proved least toxic, but a phenol derivative of this compound was most toxic.[247] Detergent poisoning symptoms in cunners were remarkably similar to those of asphyxic hypoxia.[248] Effects of Corexit on benthic and planktonic algae were dependent on several variables including the type, concentration, and mode of oil water dispersion preparation and species studied.[249, 250] Presence of a dispersant did not markedly alter the relative accumulation of Bunker C residual oil in fish tissues.[251]

Most of the studies of the effects of petroleum hydrocarbons on marine organisms were conducted in the laboratory.[252-273] In work with algae, *Chlamyclomonas* proved to be the genus most resistant to crude oil and temperature combination [252] and very heavy oils were found to have had little chemical toxicity toward juvenile *Porphyra* algae.[253] Nigerian crude oil and used crankcase oil were not as toxic as No. 2 fuel oil to green or blue-green algae and diatoms; low exposure levels sometimes promoted photosynthesis.[254]

Research with mollusks resulted in two separate papers on the modification of *Mytilus* byssal thread output by petroleum.[255, 256] Nigerian crude oil extracts suppressed nutritional efficiency in juvenile *Mercenaria* at 0.6 ppm.[257] Straughan and Hadley [258] concluded that after accounting for differences caused by temperature, one could compare toxicity data for snails from southern California and Alaska. In polychaetes, fecundity was the most sensitive reproductive parameter in a laboratory strain of *Neanthes*.[259]

Lee [260] studied crustaceans under acute and chronic exposure conditions. Fecundity among isopods was more sensitive than was growth to chronic exposure to No. 2 fuel oil. This same oil had immediate, nonreversible effects on shrimp respiration, feeding, and swimming when compared with the delayed, reversible reactions to aged oil extracts.[261] Amphipod survival was not significant in naphthalene solutions greater than 5 mg/l.[262] Crude oil water soluble fractions had only a slight effect on the carbon budget of *Crangon*.[263] Out of 18 specific compounds, indan and dimethylnaphthalene exerted the most severe effect on the swimming ability of barnacle larvae.[264] No. 2 fuel oil was the most toxic component of an artificial refinery effluent, as tested on two species of shrimp.[265] An exposure time of 24 hours to 10 mg/l crude oil produced slight increases in anaerobic metabolism of commercial shrimp.[266] From extensive laboratory testing, Laughlin and associates [267] concluded that mud crabs show ample capacity to recover from chronic sublethal exposure to No. 2 fuel oil extracts.

Research with fishes included a study that suggested that the toxicity of naphthalene was caused by its effect on blood components.[268] Separate studies revealed the effects of crude and fuel oils on fertilization processes in Baltic fish.[269, 270] Ekofisk crude exhibited only moderately disruptive effects on early development of larvae of three species, with greatest harm befalling organogenesis.[271] Continuous exposure to No. 2 fuel oil produced little effect (12.5% water soluble fraction), occasional pathological disruption of excretory and epithelial tissue (25%), or complete suppression of hatching (50%) on estuarine *Fundulus* larvae.[272] Freshwater-adapted trout were morphologically and biochemically more susceptible to the effects of Bunker C oil and a dispersant than those adapted to sea water.[273]

Daily ingestion of up to 500 mg weathered crude oil for 156 days failed to affect either testosterone metabolism in gulls [274] or growth in herring gull chicks.[275]

The often great discrepancies between tissue concentrations of hydrocarbons and apparent strength of objectionable flavor were discussed by Stansby.[276] Evidence of differing rates of metabolism for straight-chain and isoprenoid alkanes in fish muscle was pre-

sented.[277] Toluene was biomagnified to a greater extent than benzene by Pacific herring.[278] Epidermal mucus of fish was shown to be an active site of aromatic hydrocarbon uptake and metabolism.[279] Relative accumulation of benzene by young stages of herring was inversely related to age.[280] Naphthalene penetrated chorionic tissue of salmon eggs more rapidly than either benzo(a)pyrene or nC-16.[281] Melancon and Lech [282] found that diaromatic hydrocarbon release rates were directly related to brevity of initial exposure. Coho salmon accumulated low molecular weight hydrocarbons at higher relative levels and released these compounds more rapidly than flounder.[283] Elevations in detoxication enzyme systems in response to acute exposure were apparent for 1 month following induction.[284] The metabolites of dietary naphthalene were isolated from Coho salmon; temperature enhanced the retention of naphthalene.[285] Three polynuclear aromatic hydrocarbons induced both increases of hepatic aryl hydrocarbon hydroxylase activity and levels of cytochrome P-450 of trout, yet failed to change glucuronyl transferase activity.[286] Accumulation studies with invertebrates included studies with phytoplankters [287] and marine bivalve mollusks.[288-291] Skin and adipose tissues were sites of greatest accumulation of aliphatic and aromatic compounds in mallard ducks, following oral administration of these compounds.[292, 293] A comprehensive review of aromatic hydrocarbon accumulation, metabolism, and release was presented by Malins.[294]

COMPLEX EFFLUENTS AND ENVIRONMENTS

Papers dealing with the toxicity of contaminants to marine and estuarine organisms covered a multitude of tests of the effects of primarily cadmium, copper, and mercury on plants and animals. The recent work on the effects of trace contaminant toxicity tests on marine organisms is summarized in Table V according to toxicant, organism, test, and effective concentration.

As increasing numbers of tests are made on potential marine toxicants, information becomes available for more types of contaminants and organisms. Burrows and Hem [301] reviewed the work on aquatic aluminum, which covered toxicity to animals, microorganisms, and higher plants as well as the chemistry of aluminum in water, the effects of aluminum pollution in natural waters, and the prevalence of aquatic aluminum in the environment.

A few investigators have been examining the combined effects of several metals on marine mollusks. Eisler [313] found that softshell clams, *Mya arenaria*, which had been continuously subjected to a flowing raw seawater solution containing a mixture of manganese, zinc, lead, nickel, copper, and cadmium salts, could biomagnify potential environmental levels of manganese, zinc, lead, copper, cadmium, and, to a lesser degree, nickel. Oyster embryos, *Crassostrea virginica*, were used in another multiple-metal toxicity test.[323] Copper, mercury, silver, and zinc were added individually and in combination to the *Crassostrea virginica* embryos. The findings indicated that the toxicity of copper, mercury, and zinc was not influenced by 20° to 30°C temperatures, but that silver was significantly more toxic at 20°C than at 25°C. Since oyster embryo toxicity tests are conducted by several scientists, Cardwell *et al.*[315] studied the effects of filtration versus the effects of direct sampling in the subsampling of the veliger-stage larvae at the end of a bivalve mollusk toxicity; he recommended that the direct sampling method be used in future tests and that toxicant effects on mortality should be reported along with those on abnormality.

Hamilton *et al.*[317] are proponents of the trimmed Spearman-Karber method for estimating median lethal concentrations; their report compares several methods for the treatment of data from toxicity tests. Stora and Bellan [342] stated that comparisons of lethal concentrations obtained in short-term toxicity tests allow the determination of the sensitivity of species of the same ecosystem to various pollutants and the determination of those animals that might be the most susceptible to pollutants. Caution must be taken if such assumptions are to be made from the results of short-term experiments, as a study with blue crabs, *Callinectes sapidus*, indicated. The crabs' ability to avoid freshwater runoff demonstrated that laboratory and field animals do not always react in the same manner when placed in similar situations. This study found that laboratory experiments without associated field studies may cause misleading results if they are applied to a determination of impact criteria.[324]

Recently, there has been increased emphasis on full life-cycle toxicity tests, and prime candidates for such studies are mysids, in particular *Mysidopsis bahia*.[332] Other

TABLE V. The effects of trace contaminants on marine organisms.

Toxicant	Group	Species	Observed Effect	Effective Concentration (mg/l)	Reference
Aluminum	Polychaete	*Myxicola infundibulum*	80% withdrawal response	4.6×10^{-4} *M*	344
	Copepod	*Nitocra spinipes*	96-h LC50	10.0	299
Bromine	Decapod (post-larval)	*Palaemonetes* spp.	96-h LC50	0.6	302
			96-h EC50, color change response	0.5	302
	Decapod (juvenile)	*Callinectes sapidus*	96-h LC50	0.8	302
Quaternary ammonium compounds	Algae	*Chlorella* sp.	Suppression of growth	3–5 ($>10^{-5}$ *M*)	343
Cadmium	Polychaete	*Capitella capitata*	96-h LC50	5.8	335
			Suppressed reproduction	0.56	335
	Polychaete	*Ctenodrilus serratus*	96-h LC50	2.5–5.0	334, 335
			Enhanced reproduction	0.1	334, 335
			Suppressed reproduction	2.5	334, 335
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	12.1	335
			Suppressed reproduction	1.0	335
	Polychaete	*Myxicola infundibulum*	80% withdrawal response	5.2×10^{-4} *M*	344
	Polychaete	*Nereis virens*	96-h LC50	9.3	314
	Polychaete	*Ophryotrocha diadema*	96-h LC50	4.2	334, 335
			Suppressed reproduction	1.0	
	Gastropod	*Nassarius obsoletus*	96-h LC50	35.0	314
	Pelecypod	*Mya arenaria*	96-h LC50	2.5	314
	Copepod	*Nitocra spinipes*	96-h LC50	1.8	299
	Copepod	*Pseudodiaptomus coronatus*	50% reduced reproduction	0.01	333
	Copepod	*Tisbe holothuriae*	30 wk-extinction of population	0.5	319
	Mysid	*Mysidopsis bahia*	96-h LC50	0.015	332
			17-d LC50	0.0115	332
	Decapod	*Pagurus longicarpus*	96-h LC50	1.3	314
	Decapod (larvae)	*Paragrapsus quadridentatus*	96-h LC50	0.49	295
	Sea star	*Asterias forbesi*	96-h LC50	7.1	314
	Fish	*Fundulus heteroclitus*	96-h LC50	22.0	314
	Fish (juvenile)	*Morone saxatilis*	Reduction of O_2 consumption—30-d exposure	0.0005	311
Calcium	Copepod	*Nitocra spinipes*	96-h LC50	580	299
Chlorine (total)	Amphipod	*Gammarus daiberi*	1-h LC50	1.85	316
Chlorine (total residual)	Fish	*Alosa aestivalis*	15% reduction in eggs reaching the hatched stage	0.38	330
			Reduced length at hatch	~0.26	330
	Fish	*Morone americana*	15% reduction in eggs reaching the hatched stage	0.16	330
			Reduced length at hatch	0.11	330
	Fish	*Morone saxatilis*	15% reduction in eggs reaching the hatched stage	0.15	330
			Reduced length at hatch	0.051	330
Chromium (trivalent)	Polychaete	*Myxicola infundibulum*	80% withdrawal response	1×10^{-3} *M*	344
Chromium (hexavalent)	Polychaete	*Capitella capitata*	96-h LC50	5.0	335
			Suppressed reproduction	0.1	335

TABLE V—(*Continued*)

Toxicant	Group	Species	Observed Effect	Effective Concentration (mg/l)	Reference
	Polychaete	*Ctenodrilus serratus*	96-h LC50	4.3	335
			Suppressed reproduction	0.05	335
			Suppressed reproduction	~0.1	334
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	3.2	335
			Suppressed reproduction	0.05	335
	Polychaete	*Nereis virens*	96-h LC50	2.0	314
	Polychaete	*Ophryotrocha diadema*	96-h LC50	>5.0	334, 335
			Suppressed reproduction	1.0	334, 335
	Gastropod	*Nassarius obsoletus*	96-h LC50	105	314
	Pelecypod	*Mya arenaria*	96-h LC50	57	314
	Decapod	*Pagurus longicarpus*	96-h LC50	10	314
	Sea star	*Asterias forbesi*	96-h LC50	32	314
	Fish	*Fundulus heteroclitus*	96-h LC50	91	314
Cobalt	Copepod	*Nitocra spinipes*	96-h LC50	4.5	299
Copper	Dinoflagellate	*Gonyaulax tamarensis*	50% cells became non-motile	"cupric ion activity" was $10^{-10.4}$ *M*	296
	Dinoflagellate	*Gymnodinium splendens*	Reduction of growth	0.02	338
	Dinoflagellate	*Prorocentrum micans*	Reduction of growth	0.005	338
	Dinoflagellate	*Scrippsiella faeroense*	Reduction of growth	0.005	338
	Polychaete	*Capitella capitata*	96-h LC50	0.2	335
	Polychaete	*Ctenodrilus serratus*	96-h LC50	0.3	334, 335
			Suppressed reproduction	0.1	334, 335
	Polychaete	*Myxicola infundibulum*	80% withdrawal response	4.8×10^{-5} *M*	344
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	0.3	335
	Polychaete	*Ophryotrocha diadema*	96-h LC50	0.16	334, 335
			Suppressed reproduction	0.25	334
	Copepod	*Acartia clausi*	48-h LC50	0.034–0.082	328
	Copepod	*Nitocra spinipes*	96-h LC50	1.8	299
	Decapod (larvae)	*Paragrapsus quadridentatus*	96-h LC50	0.17	295
	Fish	*Clupea harengus pallasi*	Embryo mortality	0.035	336
			Larval mortality	0.3	336
Iron (ferrous)	Polychaete	*Myxicola infundibulum*	80% withdrawal response	2.2×10^{-5} *M*	344
Iron (ferric)	Polychaete	*Myxicola infundibulum*	80% withdrawal response	3.25×10^{-6} *M*	344
	Copepod	*Nitocra spinipes*	96-h LC50	21.0	299
Lead	Algae	*Ulva rigida*	Reduced O_2 production	0.1	350
	Algae	*Fucus virsoides*	Reduced O_2 production	0.1	350
	Polychaete	*Capitella capitata*	96-h LC50	11.4	335
			Suppressed reproduction	0.2	335
	Polychaete	*Ctenodrilus serratus*	96-h LC50	14.0	334, 335
			Suppressed reproduction	1.0	334, 335
	Polychaete	*Myxicola infundibulum*	80% withdrawal response	4.0×10^{-4} *M*	344
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	7.7	335
			Suppressed reproduction	3.1	335
	Polychaete	*Ophryotrocha diadema*	96-h LC50	10–20	334, 335
			Suppressed reproduction	5.0	334, 335
Tetramethyl lead	Mixed marine bacteria		Respiration rate: no effect	0.9	326
			Respiration rate: 50% change	1.9	326
	Algae	*Dunaliella tertiolecta*	Photosynthetic activity: no effect	0.45	326
			Photosynthetic activity: 50% reduction	1.65	326
	Brine shrimp (nauplii)	*Artemia salina*	Mortality: no effect	0.18	326
			Mortality: 50%	0.23	326
	Fish (larvae)	*Morone labrax*	Mortality: no effect	0.045	326
			50% mortality	0.1	326

TABLE V—(*Continued*)

Toxicant	Group	Species	Observed Effect	Effective Concentration (mg/l)	Reference
Tetraethyl lead	Mixed marine bacteria		Respiration rate: no effect	0.02	326
			Respiration rate: 50% change	0.08	326
	Algae	*Dunaliella tertiolecta*	Photosynthetic activity: no effect	0.1	326
			Photosynthetic activity: 50% reduction	0.15	326
	Brine shrimp (naupli)	*Artemia salina*	Mortality: no effect	0.025	326
			50% mortality	0.085	326
	Fish (larvae)	*Morone labrax*	Mortality: no effect	0.010	326
			Mortality: 50% mortality	0.065	326
Magnesium	Copepod	*Nitocra spinipes*	96-h LC50	720	299
Manganese	Polychaete	*Myxicola infundibulum*	80% withdrawal response	4×10^{-6} *M*	344
	Copepod	*Nitocra spinipes*	96-h LC50	70	299
Mercury	Polychaete	*Capitella capitata*	96-h LC50	<0.1	335
	Polychaete	*Ctenodrilus serratus*	96-h LC50	0.09	334, 335
			Suppressed reproduction	0.05	334, 335
	Polychaete	*Myxicola infundibulum*	80% withdrawal response	3.25×10^{-6} *M*	344
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	0.02	335
	Polychaete	*Nereis virens*	96-h LC50	0.07	314
	Polychaete	*Ophryotrocha diadema*	96-h LC50	0.09	334, 335
			Suppressed reproduction	0.1	334
	Gastropod	*Nassarius obsoletus*	96-h LC50	32	314
	Pelecypod	*Mya arenaria*	96-h LC50	0.4	314
	Pelecypod	*Rangia cuneata*	96-h LC50: 2‰ salinity	0.122	312
			96-h LC50: 13‰ salinity	0.058	312
	Brine shrimp	*Artemia salina*	Reduced life span	0.01	307
	Copepod	*Nitocra spinipes*	96-h LC50	0.23	299
	Decapod	*Pagurus longicarpus*	96-h LC50	0.05	314
	Sea star	*Asterias forbesi*	96-h LC50	0.06	314
	Fish	*Fundulus heteroclitus*	96-h LC50	0.8	314
	Fish	*Morone saxatilis*	Reduced O_2 consumption	0.005	311
Methyl-mercury	Brine shrimp	*Artemia salina*	Reduced life span	0.005	307
			Inhibition of naupli production	0.002	307
	Amphipod	*Gammarus duebani*	96-h LC50:		
			100% sea water	0.15	321
			2% sea water	0.23	321
Nickel	Polychaete	*Nereis virens*	96-h LC50	23	314
	Gastropod	*Nassarius obsoletus*	96-h LC50	72	314
	Pelecypod	*Mya arenaria*	96-h LC50	320	314
	Copepod	*Nitocra spinipes*	96-h LC50	6.0	299
	Decapod	*Pagurus longicarpus*	96-h LC50	47	314
	Sea star	*Asterias forbesi*	96-h LC50	150	314
	Fish	*Fundulus heteroclitus*	96-h LC50	350	314
Silver	Polychaete	*Myxicola infundibulum*	80% withdrawal response	2×10^{-4} *M*	344
Potassium	Copepod	*Nitocra spinipes*	96-h LC50	450	299
Zinc	Polychaete	*Capitella capitata*	96-h LC50	10.7	335
			Suppressed reproduction	0.56	335
	Polychaete	*Ctenodrilus serratus*	96-h LC50	7.1	334, 335
			Suppressed reproduction	0.5	334, 335
	Polychaete	*Myxicola infundibulum*	80% withdrawal response	1×10^{-4} *M*	344
	Polychaete	*Neanthes arenaceodentata*	96-h LC50	1.8	335
			Suppressed reproduction	0.32	335
	Polychaete	*Nereis virens*	96-h LC50	8.1	314
	Polychaete	*Ophryotrocha diadema*	96-h LC50	2.7	334, 335
			Suppressed reproduction	0.5	334, 335
	Gastropod	*Nassarius obsoletus*	96-h LC50	50	314

TABLE V—(*Continued*)

Toxicant	Group	Species	Observed Effect	Effective Concentration (mg/l)	Reference
	Pelecypod	*Mya arenaria*	96-h LC50	7.7	314
	Copepod	*Nitocra spinipes*	96-h LC50	1.45	299
	Decapod	*Pagurus longicarpus*	96-h LC50	0.4	314
	Decapod (larvae)	*Paragrapsus quadridentatus*	96-h LC50	1.23	295
	Sea star	*Asterias forbesi*	96-h LC50	39	314
	Fish	*Fundulus heteroclitus*	96-h LC50	60	314
Bleached kraft mill effluent	Amphipod	*Anisogammarus pugettensis*	50% interference with reproductive behavior	10–15% full strength	309
di-n-butyl phthalate	Dinoflagellate	*Gymnodinium breve*	96-h LC50	0.02–0.6	346
			50% growth reduction	0.0034–0.2	346
di-n-propyl phthalate	Dinoflagellate	*Gymnodinium breve*	96-h LC50	1.3–6.5	346
			50% growth reduction	0.9–2.4	346
di-ethyl phthalate	Dinoflagellate	*Gymnodinium breve*	96-h LC50	23.5–33.0	346
			50% growth reduction	3.0–6.1	346
di-methyl phthalate	Dinoflagellate	*Gymnodinium breve*	96-h LC50	125–185	346
			50% growth reduction	54–96	346

crustaceans that have been mentioned as possible test organisms for full life-cycle or chronic toxicity tests are isopods and amphipods,[298] as they are widely distributed and important in the marine ecosystem. If full life-cycle tests are not feasible, Burton and Margrey [302] have found that larval crabs, *Paragrapsus quadridentatus*, are nine times more sensitive to zinc than are the adults of the same species.

In a test that examined the effects of quaternary ammonium compounds on green algae, *Chlorella* sp., it was found that such ammonia compounds significantly suppressed plant growth at concentrations above 1×10^{-5} *M*.[343]

Wright [347–349] studied the effects of cadmium on the shore crab, *Carcinus maenus*. It was found that the accumulation of cadmium by the crab was somewhat dependent on the calcium concentration of the external medium, although apparently independent of the overall salinity of this external medium. The whole body cadmium concentration had a significant inverse correlation with the external calcium concentration, which indicated some degree of competition for binding sites between cadmium and zinc. Wright found that lowering the salinity—lowering the calcium concentration—significantly increased cadmium accumulation in the haemolymph, gills, and carapace.

In work done with the striped bass, *Morone saxatilis*, Dawson *et al.*[311] found that there was no significant difference in enzyme activity during exposure to cadmium, although the fish exhibited changes in gill tissue respiration. The lack of significant change in activity of the liver enzymes suggested a metabolism readily adaptable to environmental changes.

Tisbe holothuriae were exposed to cadmium concentrations of 0.148 and 0.222 mg/l for 30 weeks; the overall effect was the formation of U-shaped population curves, which exhibited the initial acute effect followed by a substantial recovery after about 23 weeks.[318]

The amphipod *Gammarus daiberi* was subjected to plume drifts and laboratory exposure of chlorinated power plant cooling effluent. During the plant chlorination, the amphipods survived 1-hour exposures to full-strength effluent (total chlorine concentrations of 0.05 mg/l), with no increased immediate or latent mortality. One-hour static bioassays revealed an initial LC_{50} of total chlorine to be 1.85 mg/l.[316] Further work with free chlorine and chloramine on the growth of larval lobsters, after a short toxicant exposure, provided evidence that longer lasting metabolic disturbances may result from acute exposure to chlorinated sea water. The chlorinated sea water apparently affected the energy utilization of the larval lobsters.[303] Three significant effects were noted on fish eggs and larvae exposed to chlorinated sea water. Total residual chlorine altered the hatching potential of a fish population by directly affecting numbers of eggs approaching hatch and hatching. Low concentrations of total residual chlorine (0.1 to 0.4 mg/l) affected the larval length at hatch.[330] It appeared that the toxicity of bromine chloride to post-larval shrimp, *Palaemonetes* spp.,

and juvenile blue crabs, *Callinectes sapidus*, was not affected by salinity changes as the toxicity of chlorine had been.[302] The capacity of hexavalent chromium to damage the shrimp, *Palaemonetes pugio*, was enhanced by increased temperature or decreased salinity.[315]

Batch-type experiments were conducted with copper sulfate on the dinoflagellate species *Scrippsiella faeroense*, *Prorocentrum micans* and *Gymnodinium splendens*, while a semicontinuous culture experiment was also done with *Scrippsiella faeroense*. In the semicontinuous culture the number of cells in each flask remained constant because of the daily removal of some cells and medium, with fresh medium used for replacement. The copper was added to the replacement medium and observations revealed copper to be lethal to *S. faeroense* at 0.001 mg/l, which was different from the batch cultures where copper was not lethal at 0.02 mg/l. This discrepancy could have been accounted for by the ratio of ionic copper added to the medium:cell numbers. This ratio increased in time in semicontinuous culture because copper was added every day to maintain its constant concentration in the medium, while the cell numbers remained somewhat stable because of growth and dilution. In batch cultures this ratio decreased rapidly because copper was added only once, at the beginning, and the cell numbers increased.[338] In another copper experiment with a dinoflagellate, *Gonyaulax tamarensis*, Anderson and Morel[296] demonstrated that copper toxicity was a unique function of cupric ion activity. This toxicity occurred at the calculated copper activity of natural sea waters, assuming only inorganic copper complexation. Thus, organic chelation may be necessary before *G. tamarensis* can successfully compete in the ocean environment. *Skeletonema costatum* was relatively insensitive to cupric ion activity, demonstrating no effect on growth up to copper concentrations equal to $10^{-8.5}$ *M*. Cultures inoculated from stationary phase stocks exhibited a prolongation of the log phase with increasing copper concentrations near and above the point of precipitation of the copper. The toxicity of the copper was a function of the silicic acid concentration in the medium. This effect was observed in a range of silicon tetrahydroxide concentrations above known values for the saturation of silicon uptake kinetics, thus suggesting an influence of copper on the silicate metabolism of *Skeletonema costatum*.[329]

Copper was acidentally introduced into sea water entering a circular outdoor tank used to culture pinfish, *Lagodon rhomboides*, and other marine fish. The fish were exposed for 23 hours to copper concentrations between 0.2 and 7.2 mg/l. Ninety-four percent of the fish died after being exposed to these concentrations for 10 hours. Examination revealed that serum urea, nitrogen, alkaline phosphatase, sodium, and potassium were significantly increased, and that the overall electrolyte imbalance, which had apparently resulted from impaired osmoregulation and hemolysis, was a serious terminal condition in copper-poisoned fish.[304]

Zavodnik[350] tested the influence of lead on the oxygen production by the algae *Fucus virsoides*, *Padina pavona*, *Ulva rigida*, and *Laurencia obtusa*. The exposure concentrations ranged from 0.1 to 1.0 mg/l, and the results showed that the tidal species *F. virsoides* and *U. rigida* had oxygen production slightly inhibited, whereas the other two could not adapt to experimental lead concentrations and thus decayed within 3 days.

In a study conducted with the mudsucker, *Gillichthys mirabilis*, oxygen consumption rates of the whole fish as well as that of isolated gill tissue were measured after the fish had been exposed to lead-dosed sea water for various periods of time. The results suggested that lead-induced metabolic changes may derive more from lead effects on central nervous system coordination of activity and metabolism than from direct effects of lead on intermediary metabolism enzymes in each cell.[340] Tissue-specific lead accumulation rates were determined in this same fish as a function of four variables: sea water lead concentration, duration of exposure to lead, salinity, and temperature. The spleen, gills, fins, and intestine accumulated the largest amounts of lead while the liver and muscle accumulated the least. The decay of lead from the tissues of lead-exposed fish was observed only in the gill, fins, and intestine. These three tissues all possessed an outer or inner covering of mucus. The data suggested that the turnover of lead in these tissues was a result of lead complexation with the mucus and subsequent loss of lead when mucus was sloughed off.[341]

Hrs-Brenko *et al.*[320] investigated the combined effects of lead, salinity, and temperature on the embryonic development of the mussel, *Mytilus galloprovincialis*. The data suggested that salinity changes had more effect on the embryonic development of the mussels than

did temperature and that at optimal temperature and salinity the lead nitrate effects were minimized.

The trochid snail, *Monodonta articulata,* was exposed to mercuric sulfate-dosed sea water at concentrations ranging from 0.2 to 1 mg/l. Retraction into the shell was observed as a response to the higher mercury concentrations, as was increased time periods spent above the surface. It was considered that mercury affected the snail by interfering with respiration, initially reducing interface activity, then forcing the snail out of water for longer periods of time.[339] The effects of mercury on another mollusk were examined by Nelson *et al.*,[331] who examined juvenile bay scallops, *Argopecten irradians,* after 96 hours of exposure to mercury at various salinities and temperatures. Low salinity alone enhanced the toxicity of low concentrations of mercury, as did high temperatures. High temperature and low salinity acted synergistically with mercury to increase mortality. In an experiment with the clam, *Rangia cuneata,* mercury was less toxic at lower salinity levels,[312] which was directly opposite of what was found with *Argopecten irradians.*[331]

Brown and Parsons[300] simultaneously exposed fish, *Onchorhynchus keta,* and zooplankton to three mercury concentrations—trace, 0.001, and 0.005 mg/l. Pathological effects in the fish and zooplankton appeared to coincide with saturation of metallothionein and "spillover" of mercury into the high molecular weight protein pool.

In a field study carried out in the Firth of Forth (Scotland), caged, cultured mussels, *Mytilus edulis,* were located at five locations. Mercury concentrations in the mussel tissues reached the levels similar to those found in the natural populations after about 100 days of exposure.[308]

Inman and Lockwood[321] have suggested that two compounds, methyl-mercury and lindane, interfered with the regulation of sodium when the amphipod *Gammarus duebeni* was exposed to sublethal concentrations. Both compounds disrupted the ionic-regulatory systems of *G. duebeni* sufficiently, which caused a significant fall in the concentration of sodium in the haemolymph when the animal was acclimated to 2% sea water.

In a study carried out by Kayser[323] on three dinoflagellates, *Scrippsiella faeroense, Prorocentrum micans,* and *Gymnodinium splendens,* and two diatom species, *Schroederella schroederi* and *Thalassiosira rotula,* the results showed that the multispecies exposures to zinc were a more sensitive test method for investigating the influence of zinc on planktonic algae than were monoculture experiments.

In a review of current concepts of metal metabolism with an emphasis on the factors that influence the uptake, storage, and elimination of essential and pollutant trace metals in marine organisms, Coombs and George[306] found a storage and detoxification mechanism where formation of vesicles within the cells enclose the metal within membranes. This prevented contact of excess metal with vital constituents and effectively detoxifies it until eliminated or passed onto other tissues. Karnaukhov *et al.*[322] have found a relationship between content of carotenoids in bodies of marine mollusks and their tolerance to environmental pollution. It was shown that the mollusks with high carotenoid content in their bodies have a high resistance to environmental pollution.

Davis *et al.*[310] have designed marine ecosystem testing units for integrating responses of communities of marine organisms to low-level pollutants under natural conditions of temperature, weather, light, and season. The units have demonstrated responses to chlorination at levels below chemical detectability.

Moore[327] compiled a synthesis of available knowledge of inorganic particulate suspensions in the sea and their effects on marine animals to emphasize where data are deficient and to ensure the ready accessibility of existing knowledge.

SLUDGE, WASTEWATER, MINING, AND DREDGING

The status of ocean dumping of wastewater and sludge was reviewed in the New York Bight and the southern California Bight.[351] The health effects of wastewater discharge were reviewed for the Mediterranean Sea.[352] In the New York Bight, Swanson[351] concluded that sludge dumping was not a significant factor affecting nearshore water quality and that discharges into the Hudson River must first be controlled.

Officer and Ryther[353] concluded that diversion of primary wastewater discharges to deep water sites is a more effective method for controlling nearshore and estuarine pollution than conversion of nearshore discharges to secondary treatment. Wastewater sludge decomposes at similar rates at 4° or 20°C; decomposition is enhanced by contact with bottom

sediments.[354] Morton [355] reviewed methods for monitoring sludge dumping at sea, and Ayres [356] found fecal bacteria to be a useful tracer for mapping sludge deposits on the sea floor.

A small, intertidal primary treated discharge at a southern California island changed community structure by enhancing the growth of a toxicant-tolerant suspension feeding food web, which was high in energy content.[357] Food-processing effluents caused major changes in benthic infauna in the Dutch Wadden Sea,[358] and inshore sandy beach communities responded to diversion of wastewater in South Africa.[359]

Tailings from copper mines in Chile silted an adjacent harbor, produced fine sludge-like material on beaches, and reduced secchi depths by 80% over nearly 10 km of coastline.[360] Rees [361] reviewed likely ecological impacts of manganese nodule mining at sea.

Rhoads *et al.*[362] concluded from a study in Long Island Sound that dredge spoils can be managed in ways that enhance productivity rather than diminish it in deeper estuarine waters. In shallow waters of Chesapeake Bay, Pearson and Bender [363] found that effects of dredge spoils dumping on marine benthos were still obvious after Tropical Storm Agnes. Dredge spoils also caused acute and sublethal effects on coral reefs.[364] Feder *et al.*[365] documented a variety of debris from foreign and domestic fishing fleets lying on the floor of the Bering Sea.

Suspended solids concentration above 1 000 mg/l reduced hatching success in white perch and striped bass, whereas concentrations above 500 mg/l reduced larval survival of the two species [366]; concentrations higher than 100 mg/l affected larval survival in American shad. Considering measured concentrations and duration during dredging, the authors concluded that excess solids from dredging were not a significant acute hazard to fish eggs and larvae. Sullivan and Hancock [367] concluded that there is no justifiable reason to conduct zooplankton dredge effects studies in ocean-dominated estuaries, while systems with endemic zooplankton populations should be investigated in detail.

Harding and Phillips [368] determined that PCBs attached to particulates, including those in secondary treated effluent, are rapidly transferred to phytoplankton in sea water. Young and Heesen [369] also found that DDT and PCBs in contaminated marine sediments were passed through the food web while halogenated benzenes from municipal wastewater were less persistent in marine sediments and even less so in benthic organisms. Trace metals from wastewater discharge are taken up by lower trophic level sea food organisms (shellfish), but not by marine fish.[370] Eganhouse and Young [371] concluded that despite major sediment contamination, organic and total mercury were not contaminating tissues of larger benthic fishes and invertebrates.

Use of small otter trawls for assessing fish and fish health at coastal dumpsites was reviewed by Mearns and Allen.[372] Optical Emission Spectrometry was shown to be a rapid, cost-effective method for simultaneously measuring a large suite of heavy metals in marine organisms.[370]

DISEASES

Diseases of known and unknown origin in marine animals continue to be reported. Work by a number of authors [373-380] indicated that pollutants and other sources of environmental stress can lead to increased growth of infectious microorganisms and increased infections in fishes and invertebrates. Giles *et al.*[373] confirmed that exposure to crude oil leads to a rapid infectious fin rot disease in mullet. Viral infections and hamartomas increase in commercial shrimp exposed to pollutants.[374, 376] Detergents and phenols also caused tissue alterations in isopods [378] and pelecypods.[377] Hodgins *et al.*[379] concluded that there is no direct evidence that petroleum is an etiological factor in tumor diseases, but that petroleum can suppress immune responses, leaving organisms open to increased infection. Colwell *et al.*[380] hypothesized that vibrio bacteria, long known to be virulent pathogens under some conditions, are the dominant bacterial species in brackish, estuarine, and coastal waters, are ecologically important in mineralizing organic matter and chitin, and contain at least three congeneric species groups (including *Vibrio cholera*) that replace one another as ecotypes.

Neoplastic diseases occur in populations of estuarine clams on the U. S. coast, but the cause of the abnormalities has not been confirmed.[381, 382]

Vandorpe and Smigielski [383] provided radiographic evidence that trace metals in cod and flatfish are concentrated in cellular-size nodes in liver tissue and are not evenly distributed either intracellularly or extracellularly. Lambert [384] used *Salmonella typhimurium* strains to identify mutagens in shellfish. Moshiri

et al.[385] provided evidence that algal metabolites, rather than low dissolved oxygen, can be implicated in estuarine fish kills.

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